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IRRIGATION: ITS IMPLICATIONS WITH RESPECT TO GROUND WATER QUALITY

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EXTENDED ABSTRACT

The portion of irrigation water that is not intercepted by crops, but returns to a water system as surface or subsurface flow, is known as irrigation return flow. Irrigation return flow, and its various properties may significantly affect the quality of ground water.

In order to evaluate the implications of irrigation return flow with respect to ground water, an understanding of the principles which govern ground water movement and quality is necessary. Factors such as ambient ground water quality, soil characteristics, and ground water flow rates must be considered in conjunction with the mobility and rate of attenuation of specific contaminants. Once pollution of ground water takes place, the remedial measures are costly.

The amount of available ground water greatly exceeds that of surface water on the Canadian Prairies. However, the quality and quantity of available ground water is highly variable.

Southern Alberta has made the largest commitment to irrigation within the Prairie region. Saskatchewan and Manitoba follow in that order. Most irrigation takes place within areas of Brown and Dark Brown soil groups.

When irrigation water is applied, salts will tend to accumulate in the upper soil profile as water is lost through evaporation and transpiration. In order for soil to retain its fertility, salts must be removed through leaching. Recent research has demonstrated that a large net transfer of salt to ground water, through heavy leaching, may not degrade the ground water quality as significantly over the long term as would a lower leaching fraction and hence higher concentration of salt. The hydrologic conditions, the quality of the irrigation water and receiving water, and the presence of precipitable salts in the soil determine the extent to which this is true.

Effluent irrigation is increasingly being used as a method of urban sewage disposal. Within the prairies there are a number of such projects. The principal contaminants from effluent irrigation which could affect ground water are salts and nitrates.

Ground water is not used extensively in Alberta and Saskatchewan as a supply of irrigation water. With the exception of a few surficial unconfined aquifers, ground water supplies are inadequate to meet irrigation demands. In aquifers capable of yielding sufficient water, the quality of the water is too poor to allow sustained irrigation. Thus, the use of ground water for irrigation supply in these two provinces will probably not increase much in the future. The presence of a few high-yielding, high-quality unconfined aquifers in southern Manitoba has made ground water a viable irrigation supply in that area.

Irrigation may cause accelerated movement of pesticides to the ground water table. The degree to which a given pesticide may contaminate ground water is dependent upon its sorption capacities, solubility, and persistence. Soil properties such as weight percent organic matter, permeability, and soil moisture content also affect the ability of pesticides to move through the soil zone.

The principal herbicide applied in western Canada is 2,4-D. It accounts for almost one-half of the total herbicide applications within the region. Dicamba (another common pesticide) has the greatest potential to cause ground water contamination. Treated irrigation ditches, which receive large amounts of herbicides, are another source of ground water contamination. In such cases, Bromacil is the greatest threat.

Contamination of aquifers by nitrogen fertilizer is common in the world's agricultural areas, especially where irrigation is heavily practised. Use of artificial fertilizers, particularly nitrogen fertilizers, has increased a great deal over the last twenty years in Canada and the United States. Ground water contamination from this source can be averted to a large extent by careful management of the application of both fertilizer and irrigation water.

RÉSUMÉ ALLONGÉ

La partie des eaux d'irrigation qui n'est pas interceptée par les cultures, mais retourne dans un réseau hydrographique sous forme d'écoulements superficiels et souterrains, s'appelle l'eau d'irrigation restituée. Cette eau restituée et ses diverses propriétés peuvent influer considérablement sur la qualité des eaux souterraines.

Pour évaluer le rôle de l'eau d'irrigation restituée dans les eaux souterraines, il faut comprendre le principes que régissent le mouvement et la qualité des eaux souterraines. Il faut examiner des facteurs comme la qualité des eaux souterraines. Il faut examiner des facteurs comme la qualité des eaux souterraines environnantes, les caractéristiques du sol et les débits des eaux souterraines de concert avec la mobilité et le taux de diminution de contaminants particuliers. Une fois les eaux souterraines polluées, les mesures correctives sont couteuses.

La quantité des eaux souterraines disponibles est de beaucoup supérieure à celle des eaux de surface dans les Prairies canadiennes. Cependant, la qualité et la quantité des eaux souterraines disponibles sont très variables.

Dans la région des Prairies, c'est dans le sud de l'Alberta que l'irrigation est la plus répandue. Puis suivent la Saskatchewan et le Manitoba. L'irrigation est concentrée dans le régions des groupes de sols bruns à bruns foncés.

Pendant l'irrigation, les sels tendent à s'accumuler dans la couche supérieure des sols à mesure que l'eau est éliminée par évaporation et transpiration. Pour que le sol demeure fertile, les sels doivent être éliminés par lixiviation. Une étude récente a démontré qu'il est possible qu'un important transfert net des sels dans les eaux souterraines contamine moins à long terme les eaux souterraines qu'une faible fraction de lixiviation et donc qu'une concentration élevée de sels. Les conditions hydrologiques, la qualité des eaux d'irrigation et des eaux réceptrices, et la présence de sels précipitables dans le sol déterminent à quel point cela est vrai.

L'irrigation par des effluents est une méthode de plus en plus répandue d'élimination des eaux d'égout urbaines. Dans les prairies, il existe plusieurs projets de ce genre. Les principaux contaminants des effluents d'irrigation qui pourraient dégrader les eaux souterraines sont les sels et les nitrates.

Les eaux souterraines sont peu utilisées en Alberta et en Saskatchewan pour l'irrigation. l'exception de quelques aquifères superficielles non confiées, l'approvisionnement en eaux souterraines est insuffisant pour répondre aux besoins d'irrigation. Dans les aquifères assez productives, la qualité de l'eau n'est pas assez bonne pour assurer une irrigation soutenue. Par conséquent, l'utilisation des eaux souterraines à des fins d'irrigation dans ces deux provinces ne se développera probablement pas dans le futur. La présence de quelques aquifères non confiées, très productives et de grande qualité, dans le sud du Manitoba a rendu viable l'utilisation des eaux souterraines à des fins d'irrigation dans cette région.

L'irrigation peut causer un mouvement accéléré des pesticides dans la nappe d'eau souterraine. Le degré de contamination des eaux souterraines par un pesticide donné dépend du pouvoir sorbant, de la solubilité et de la persist-

ance de ce pesticide. Les propriétés pédologiques comme le pourcentage en poids de matières organiques, la perméabilité et la teneur en eau influent également sur la capacité des pesticides de se déplacer dans le sol.

Le principal herbicide appliqué dans l'Ouest canadien est le 2,4-D. Il représente près de la moitié de la quantité totale d'herbicides appliquée dans la région. Le Dicamba (autre pesticide courant) est le pesticide le plus susceptible de contaminer les eaux souterraines. Les fossés d'irrigation traités, que reçoivent de grandes quantités d'herbicides, sont une autre source de contamination des eaux souterraines. Dans ce cas, la plus grande menace provient du Bromacil.

La contamination des aquifères par les engrais azotés est fréquente dans les régions agricoles du monde, notamment dans les zones d'irrigation intensive. L'utilisation d'engrais artificiels, en particuliers les engrais azotés s'est beaucoup répandue au cours des vingt dernières années au Canada et aux États-Unis. La contamination des eaux souterraines par cette source peut être en grande partie évitée par une gestion judicieuse de l'utilisation des engrais et des eaux d'irrigation.

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1. INTRODUCTION

1.1 Overview

The irrigation of crops to supplement annual precipitation during the growing season is becoming more widely practised in western Canada. This is fostered by greatly improved yields and financial returns realized by irrigating crop land. An example of this is found near Lake Diefenbaker in Saskatchewan where yields of 10 to 15 bushels per acre (1 to 2 tonnes per hectare) of wheat under "*dryland*" conditions rise to consistent yields of 50 to 60 bushels per acre (0.6 to 0.7 tonnes per hectare) when irrigated (Cashman, 1982). It has been suggested that the revenue for the farmer could increase by almost a factor of six as a result of improved yields (Smith, 1978).

The impact of irrigation upon ground water can be described in terms of: (1) a rise in the local water table, (2) changes in the ground water flow regime, and (3) alteration of ground water quality. These phenomena are especially significant in the Prairie Provinces, where the climate is arid, and a large portion of the annual hydrograph in some streams is baseflow (derived from ground water).

Kharchenko and Maddock (1982) provide examples from India, Australia, and Colorado (United States) of dramatic changes induced through irrigation in both water table elevation and ground water flow. In the Indus Plain of India, ground water rise has varied from 0.5 to 1.0 metres per year since irrigation was introduced at the beginning of the century. It has been estimated that 35 per cent of India's irrigated land is seriously saline, with subsequent damage to arable lands due to waterlogging and salinity increasing at a rate of 20,000 to 40,000 hectares per year. An almost 50 metre rise in elevation has been recorded over the last 90 years in Australia's Murray Basin. Salt balance studies in the Murray River have shown an increase in annual salt load of 850,000 tons, of which 500,000 tons are demonstrated to be a result of ground water discharge. Miller (1980) states that, in the United States, irrigation practices constitute "*. . . a major problem which has led to a number of areally extensive ground water contamination cases.*"

It is the intent of this discussion to examine the potential contamination of ground water as a result of irrigation in the context of the Canadian prairies (Alberta, Saskatchewan and Manitoba). The major issues dealt with here are salinity, sewage effluent, pesticides, and fertilizers. This paper is intended to serve only as a cursory review of the topic. A summary the factors to be considered, along with discussion of supporting studies, will be presented in each case. It is hoped that this report will serve as a contribution to future and ongoing studies aimed at preventing or mitigating the impact of irrigation return flows on ground water quality.

2. CONTROL FACTORS

2.1 Ground Water Principles

Beneath the soil surface lies the unsaturated zone, or zone of aeration. Here water and gases are contained in the pores between soil particles. The water is at less than atmospheric pressure while the gases are generally at atmospheric pressure (U.S. Water Resources Council, 1980). Within the zone of aeration is the crop root zone which extends from the soil surface to the maximum potential rooting depth for the crop in question. The zone of aeration is bounded on the bottom by the water table, which marks the beginning of the zone of saturation. This zone is under greater than atmospheric pressure and all existing spaces are saturated with water.

2.1.1 Ground Water Flow

In general, ground water flows from a recharge area to a discharge area. In unconfined conditions, the slope of the water table indicates the direction of flow, which is from areas of higher to lower water table elevation (Freeze and Cherry, 1979). The hydraulic gradient refers to the slope, or change in static head per unit of distance in some direction (U.S. Water Resources Council, 1980). Ground water generally moves slowly, and its flow is often measured in centimetres per day or per year. The subsurface geological formation which transports, stores, and is capable of yielding ground water of significant quantity, is called an aquifer. The rate of flow of ground water depends both on the permeability of the material and on the hydraulic gradient in the aquifer. The geometry of the spaces through which it flows is the most significant factor affecting the permeability of the material (Gibb and Jones, 1974). The degree of sorting of the material also plays a crucial role in the flow of ground water. When there exists a wide range of particle sizes (poorly sorted material), the permeability of the medium tends toward the permeability of the smallest particles (Gibb and Jones, 1974). The velocity of flow can be expressed by Darcy's Law, which states that velocity equals permeability, multiplied by head loss, divided by the length of the flow path. Head loss is a measure of the difference in height to which ground water rises in a well or tube at two points along its flow path.

2.1.2 Aquifer Types

There are three basic aquifer types: the unconfined, the confined, and the perched. In an unconfined aquifer, the water table level fluctuates with the amount of incoming moisture. If a well is drilled into an unconfined aquifer, the water will rise to the height of the water table. In a confined aquifer, the height above some datum to which water will rise in a well will exceed that of the top of the aquifer. A perched aquifer (or water table) is found where an impermeable layer intercepts the percolating water and contains it above the main zone of saturation.

2.1.3 Ground Water Quality

Ground water is generally somewhat better shielded from surface input of anthropogenic contaminants than are surface waters, owing to the presence of overlying geologic materials. As a result, ground water is normally free of such contaminants as synthetic compounds and microbes. The dissolved oxygen content is usually very low, measurable only at shallow depths in recharge zones. Its chemical composition and temperature are nearly constant.

In consolidated rock, the transport of contaminants is affected by two types of permeability: intergranular, and fracture. With intergranular permeability, contaminants tend to move from the input location in a broad front, undergoing dispersion and dilution along the direction of flow. If movement can only take place through fractures, the contaminant will tend to move more rapidly. Dispersion and dilution will again occur predominantly in the main direction of travel, that is along the direction of dominant fracture permeability. However, under the effects of fracture permeability, mixing will generally be quicker and dispersion more pronounced.

Intergranular permeability is dominant in sedimentary rocks, which are made of distinct particles (such as sandstone, conglomerate, and sandy siltstone). In this condition, microscopic dispersion takes place through mixing caused by the flow of water around individual grains (Everett, 1980). Fracture permeability may be dominant in igneous rocks (granite, basalt), chemical rocks (limestone, gypsum), metamorphic rocks (slate, quartzite), and in some sedimentary rocks (shale, siltstone, and sandstone) (Gibb and Jones, 1974). Mixing which results from the movement of ground water through geological formations of varying permeability is known as macroscopic dispersion (Everett, 1980).

Among unconsolidated materials, clean gravel and sands have the highest permeabilities and favour rapid contaminant transport. The next most permeable are fine sands, fractured glacial tills, silts, and mixtures of sand, silt and clay. These are followed by stratified clays and finally unweathered clays (Gibb and Jones, 1974). In general, the size, abundance, and connection between pore spaces and fractures determines the rate at which the ground water and its constituents travel through an aquifer.

Once ground water is contaminated, it usually remains so for long periods of time. One reason for this is the slow flushing rates and potentially long residence times of ground water in an aquifer. Another reason is the slow rate of breakdown or degradation of a contaminant in a ground water reservoir. This is a result of the low concentrations of dissolved oxygen (DO), constant temperatures, and the relative absence of biological activity. Compared with surface waters, the laminar flow of contaminanted ground water leads to relatively little mixing with uncontaminated water. The concentration of the contaminant is, however, attenuated in the ground water zone through the physical processes of diffusion and dispersion, and the chemical processes of ion exchange transformation and sorption.

Any attempts to remove contaminants from ground water are extremely costly. Jackson (1982) outlines three categories of remedial measures that can be used to address ground water contamination problems. The method most often used consists of pumping or withdrawing contaminated water from the aquifer and treating it. A second category involves creating some sort of impermeable barrier to ground water flow. A third category of remedial measures is the *in situ* treatment of ground water through the injection of chemical or biological reagents. In any case, it appears that the most cost-effective solution to ground water contamination is prevention.

2.2 Irrigation Methods within Western and Northern Region

The three main methods of irrigation are flood, sprinkler, and drip irrigation. In 1982, about half of the systems in use in the Prairie Provinces were of the flood irrigation type, and half were of the sprinkler type (Jame, pers. comm., 1982). The number of sprinkler systems has increased more rapidly in recent years than the number of flood irrigation systems, primarily due to greater versatility and reduced labour requirements.

2.2.1 Flood

With flood (also called surface, or gravity) irrigation, water is applied directly onto the land from ditches through gates or siphons. The water spreads over the field under the force of gravity. Extensive levelling and working of the land is required to ensure even coverage of the applied water. Two common types of surface irrigation are known as border ditch and border dyke. In the first method, ditches are placed in a parallel position 20-35 metres apart. The border dyke method uses parallel dykes 10 to 15 cm high and spaced 10 to 20 metres apart to guide and contain applications of water (Bachman *et. al.*, 1980). A study at Swift Current, Saskatchewan, revealed no significant difference between the water application efficiency and uniformity of these two types of irrigation. They are both suited to land with slopes of less than two per cent (Korven and Randall, 1975).

Two other kinds of flood irrigation are the furrow and corrugation methods. With each of these types, only partial flooding of the field occurs. Small channels, usually spaced one metre apart, are intermittently filled with water to irrigate row crops. These can be used on land with a slope of up to 5 per cent (Korven and Randall, 1975). Corrugation flooding on sloped terrain, can result in soil loss from irrigated fields due to water erosion.

Basin irrigation is an additional variation of flood irrigation. A nearly level plot of land is enclosed by dykes which retain any applied water until the water infiltrates the soil.

2.2.2 Sprinkler

Sprinkler irrigation is also widely used in western Canada. Two common variations are the side-roll and center-pivot. The side-roll system is a hollow pipe (lateral) usually 300 metres long. It is mounted on large steel wheels spaced about 12 metres apart. Rotating sprinklers are also spaced apart evenly on the lateral. In the center of the unit is a small engine which is used to move the lateral to the next irrigation set. The center-pivot system is considerably more expensive than the side-roll, but is much less labour intensive (Jame, pers. comm., 1982). Like the side-roll, this system is also 300 metres long. The major difference is the fact that the centre-pivot propels itself in a circle. Other sprinkler methods that are used less extensively include boom sprinklers, hand-move towlines, travelling and "*square-matic*" sprinklers (Korven and Randall, 1975).

Despite its greater costs, sprinkler irrigation is becoming increasingly popular because it has several advantages over surface irrigation. A sprinkler system is adaptable to both level and sloping ground and can hence be used on land unsuitable for flood irrigation (Bachman *et. al.*, 1980). The flood systems are not recommended for soils with either high intake or very low intake rates (Environment Council of Alberta (ECA), 1982). Sprinkler systems use water more efficiently and effectively as water is distributed in more uniform and controlled amounts (Korven and Randall, 1975). During windy periods, however, the uniformity of application is very poor. Unlike surface irrigation, the amount of water applied with sprinkler irrigation can be effectively controlled by making adjustments in the duration of application corresponding to climatic conditions. Consequently, sprinkler irrigation results in less leaching and less percolation below the root zone. With surface irrigation, the application of additional water to ensure adequate moisture applications to high areas or areas of low intake rates in the irrigated field can result in a large amount of water leaching below the root zone (Bachman *et. al.*, 1980).

2.2.3 Drip

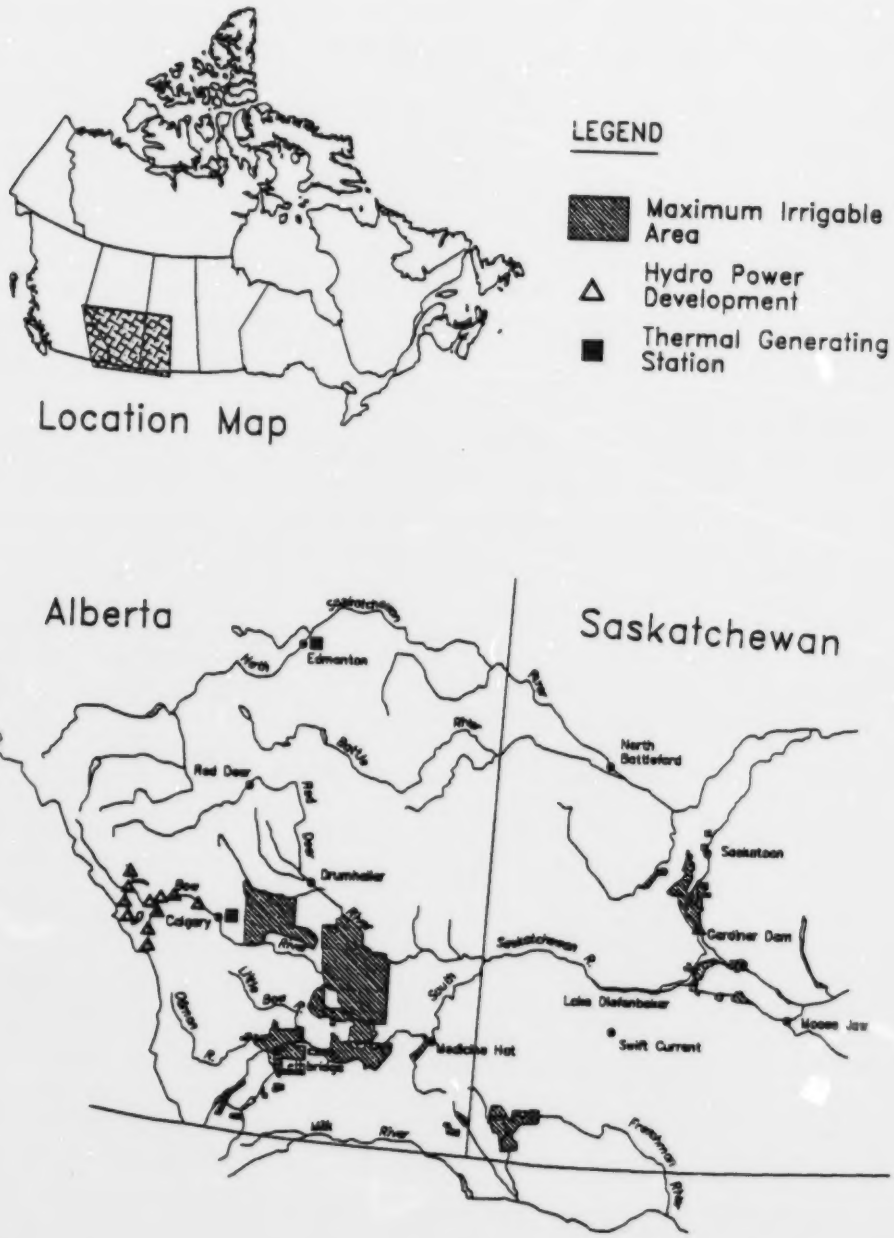
A third category of irrigation is known as drip, or trickle, irrigation. This type utilizes perforated tubing to dispense the water and is best suited to row crops and sites where supply water quality is quite good, because drip systems are prone to clogging. Since the cultivation of row crops is not common in the region, it is not widely practised.

2.3 Location of Irrigation on the Prairies

2.3.1 Alberta

In Alberta, the first major irrigation projects were developed in the late 1800's, financed by private companies and constructed by railroad companies or land development corporations. Most of the 13 irrigation districts currently existing were established between 1915 and 1968 (Lane and Sykes, 1982). These are located in the south and south-central regions of the

Figure 2.1: Irrigation Areas: ***Approximate location of public irrigation districts for Alberta and Saskatchewan. (based upon 1988 information supplied by Water Planning and Management Branch, IWD, Regina, and Saskatchewan Water Corporation, Moose Jaw)***



Province (Figure 2.1) and are run by a Board of Directors which charges an annual rate for water users. Cost-sharing agreements between the province and the districts are administered by an Irrigation Council (Lane and Sykes, 1982). The actual amount of land irrigated (and corresponding water consumption) varies dramatically from year to year depending on the precipitation received. The Prairie Farm Rehabilitation Administration (PFRA, 1982) estimates that an average of 70 per cent of the licensed land is actually being irrigated in any one year.* In 1978, 311,820 ha were actually irrigated in the districts (PFRA, 1982). In comparison, approximately 60,700 ha were irrigated in private irrigation projects in 1978. The majority of these are located in the Oldman, South Saskatchewan, and Red Deer River basins (W-E-R Engineering, 1982) (Table 2.1 and Figure 2.1).

Table 2.1: Irrigation in Alberta by Drainage Basin — 1978

Drainage Basins	Private (ha)	District (ha)
Oldman River Basin	16,135	110,190
South Saskatchewan River	16,035	53,900
Bow River	10,808	65030
Red Deer River	10913	82,700
Milk River	2,324	—
North Saskatchewan River	4,711	—
Athabasca River	504	—
Beaver River	84	—
Peace River	196	—

Source: PFRA, 1982

2.3.2 Saskatchewan

The first irrigation projects in Saskatchewan were developed in the Cypress Hills area during the 1890s. Since 1936, the PFRA has developed six major irrigation projects with a total area of 9,230 ha. In addition, there are currently 13 Water User Districts (W.U.D.) in the province, established under provincial statutes. These are operated by water user district associations and account for approximately 4,880 ha of irrigable land. Estimating land actually irrigated at 70 per cent brings this figure down to 3,416 ha.

The largest irrigation project in Saskatchewan was part of a joint federal-provincial agreement known as the South Saskatchewan River Irrigation Project (S.S.R.I.P.). This project began in 1966 and by 1978 approximately 11,450 ha were actually irrigated. The estimated potential development is 65,000 ha.

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* For purposes of uniformity the PFRA 1982 study data will be used. Other relevant reports are Prairie Provinces Water Board (1982) and W-E-R Engineering Ltd. (1982).

The amount of private (non-district) lands irrigated in the province in 1978 was approximately 51,800 ha, with the bulk located in the South Saskatchewan and Missouri River Basins. This reveals the importance of non-district irrigation in Saskatchewan. In the future, however, district irrigation will likely surpass private through implementation of the Canada/Saskatchewan Subsidiary Agreement on Irrigation-Based Economic Development, which was signed in 1986, and covers construction of the Luck Lake (8,300 ha.) and Riverhurst (8,700 ha.) phases of the South Saskatchewan River Irrigation Project (Table 2.2 and Figure 2.1).

TABLE 2.2: Irrigation in Saskatchewan by Drainage Basin - 1978

Drainage Basins	Private (ha)	District (ha)
South Saskatchewan River	18,700	30,710
Missouri River	9,900	7,300
Old Wives Lake	8,500	4,610
Qu'Appelle River	6,000	110
North Saskatchewan River	4,600	—
Souris River	1,700	62
Assiniboine River	400	—
Saskatchewan River	800	—
Churchill River	1,200	—

Source: PFRA, 1982

2.3.3 Manitoba

Manitoba receives more precipitation than Alberta or Saskatchewan. As a result, irrigation is less developed than in the other two provinces. There are no government irrigation projects in Manitoba (Klassen, pers. comm., 1982). The bulk of the private irrigation projects (71 per cent) are located in the Assiniboine River Basin. The remainder of irrigation is spread evenly between the Souris, Pembina, and Red River basins and the Lake Manitoba Basin. In 1978, approximately 4,575 ha were licensed for irrigation. In 1979 this had risen to 5,990 ha and in 1980 an additional 3,200 ha was developed for irrigation (PFRA, 1982), indicating the rapid expansion of irrigation in this province. Table 2.3 shows the areas under irrigation in Manitoba by drainage basin.

Table 2.3: Irrigation in Manitoba by Drainage Basin - 1978

Drainage Basins	Privately Irrigated (ha)
Souris River Basin	245
Assiniboine River Basin	3,280
Pembina River Basin	365
Red River Basin	365
Lakes Basin	325

Source: PFRA, 1982

2.4 Use of Ground Water For Irrigation

Ground water accounts for 40 per cent of the irrigation supply in the United States (Senft and Solomon, 1986). On the Canadian prairies, however, ground water usage is much more limited, and most irrigation currently taking place is done using surface water (Jame, pers. comm., 1982).

2.4.1 Quality Constraints

The ground water in Saskatchewan typically has a high salt content (Bachman *et al.*, 1980). The high yield aquifers (yield of 28 L/s or greater) generally have total dissolved solids (TDS) in the range of 1,000 to 3,000 mg/L with isolated samples reaching TDS levels as high as 6000 mg/L (Saskatchewan-Nelson Basin Board (SNBB), 1972). Most of the high yield aquifers in Alberta and Manitoba have a lower TDS (less than 2000 mg/L) than those found in Saskatchewan (SNNB, 1972).

Ground water with a high salt content requires that a correspondingly greater amount of water be applied to maintain a sufficiently low concentration of salts in the root zone (The implications of this are discussed in Section 3.1 of this report). With proper management practices it is possible to maintain an acceptable soil salinity profile. The sodium content of the ground water used for irrigation is the primary factor limiting its suitability for this purpose. "*High sodium in the irrigation water can drastically reduce the rate of water movement in soil as a result of ion exchanges in the soil complex*" (Jame, 1981). The term "*sodium adsorption ratio*" (SAR) is used to predict the degree to which irrigation water tends to enter into cation-exchange reactions in soil. High values indicate a high potential for sodium in the applied water to replace calcium and magnesium in the soil (Hern, 1970), and to cause dispersion of clay particles within the soil, which can lead to a reduction in the permeability of the soil and reduces the infiltration of water (Clifton and Associates, 1987). An adjusted SAR of 10 to 16 is considered high; Jame and Nicholaichuk (1981) indicate that two-thirds of the ground water in Saskatchewan has a sodium absorption ratio above 16. Clifton and Associates (1987) indicated that of 40 major aquifers investigated, 15 had an average adjusted SAR over 16. Bachman *et al.* (1980) concluded that much of the available ground water in Saskatchewan is of marginal quality for

irrigation and if used for an extended period for irrigation, would cause salinity problems to develop in most soil types. The sodium content of ground water is also of some concern in Alberta and Manitoba (Bachman *et al.*, 1980).

In addition to the consideration of irrigation water quality in the management of irrigated soils, it is recognized by some researchers that poorer quality irrigation water may be safely used on lighter, more granular soils, but not on heavier more clayey soils. In addition to textural considerations, the type of clay minerals present will also affect the resistance of a soil to structural breakdown, decreased permeability, and salt buildup. According to Cameron (1988) however, the present state of knowledge is not advanced enough to justify inclusion of soil texture as an additional consideration in irrigation water quality guidelines. Further research is necessary to successfully incorporate soil texture, and other factors such as water table depth, stratigraphy (depth to impermeable layers), and clay mineralogy into water quality/soil suitability guidelines.

2.4.2 Alberta

In Alberta there are no sizeable ground water irrigation projects. Communication from Vern A. Carlson, Groundwater Rights Branch, has indicated that the number of projects in Alberta utilizing ground water are few and very small in terms of water used (Bachman *et al.*, 1980). This is confirmed by Hess (1986), who estimated that only 0.1 % of water used for irrigation in Alberta is ground water.

2.4.3 Saskatchewan

There are currently three irrigation projects in Saskatchewan using ground water (Bachman *et al.*, 1980; Gillies, pers. comm., 1982). One project is located near Nokomis, Saskatchewan. The project went into operation in the spring of 1980 when 67 ha were irrigated by pumping from the Hatfield Valley Aquifer. In 1981 another 49 ha were irrigated with center-pivot, and recently another parcel of land was put under irrigation using side-roll. The water is slightly brackish (TDS = 2,000 mg/L; specific conductance 4100 uS/cm) and has a SAR value of 15.4. The second project is located near Fox Valley, Saskatchewan. It has been in operation since 1977 and the ground water is of good quality with a low SAR and TDS. The third project, which is located near Donovan, Saskatchewan, went into operation in 1981. Two center-pivot irrigation systems are used on 130 ha of land. The TDS concentration of the ground water has been measured at 2,430 mg/L, the specific conductance is around 3200 uS/cm, and the SAR is 13.5. The Nokomis and Donovan irrigation projects are experimental, and the results have been summarized by Gillies, (1987). Both projects have seen an increase in the amount of salt accumulated in the soils, but this has not appeared to affect crop production after five years of irrigation. However, Gillies (1988) states "*preliminary results of this long-term study suggest that unless the soils being irrigated are light and surface profiles are deep, harmful levels of soluble salts will accumulate within the root zone. Significant impacts on the quality of shallow ground water and on the physical characteristics of the source aquifer have not been observed.*"

Franc (1985) has estimated that 71,000 ha of land in Saskatchewan has the potential to be irrigated with ground water. Roeper and Rey (1986) analyzed 750 water quality records from 31 major aquifers, and found that drift and buried valley aquifers are generally the best ground water supply to irrigation, while fewer than 20 percent of bedrock aquifers offered a suitable supply. The Cypress Hills and Wynyard Formations produce water suitable for irrigation, while the Ravenscrag and Eastend Formations are generally unsuitable as a supply.

2.4.4 Manitoba.

In Manitoba, ground water has been utilized for irrigation much more extensively than in either Saskatchewan or Alberta. At present more than 115 private ground water irrigation developments exist in Manitoba on a total acreage of over 6,000 hectares. Approximately 65 per cent of this land is located near the town of Carberry, with the water being drawn from the Assiniboine Delta aquifer (Swift, pers. comm., 1982). A study conducted by MLM Ground Water Engineering (1979) on the quality and quantity of supply from the Assiniboine Delta Aquifer near Shilo indicated that the ground waters were of excellent quality for irrigation, with SAR values less than 0.3.

2.5 Physical Factors

2.5.1 Aquifers

Ground water is vastly more abundant than surface water within the Prairies. While highly variable in terms of quality, ground water serves as the main source of supply for virtually all rural water use other than irrigation. The quality and quantity of available ground water is dependent upon the location, extent, and type of aquifers.

An aquifer is a saturated geologic unit capable of yielding significant quantities of water. This definition may be expanded to include quality considerations. Thus the aquifer as newly defined should be capable not only of yielding significant quantities of water for a particular use, but also must yield water that is of suitable quality for that use. Within the Prairie Provinces, aquifers can be generally classified as surficial, or bedrock.

Surficial Aquifers

Surficial aquifers include intertill deposits of sand and gravel, buried preglacial valley alluvium lying directly on the bedrock surface, sheet deposits of glacial outwash, and Recent alluvium in hydraulic contact with present-day rivers or streams. These deposits are almost always unconsolidated. Sustainable well yields vary greatly between the different types of surficial aquifers, being dependent upon the grain-size and degree of sorting of the sediments (hence the permeability), and the aquifer geometry (its thickness and areal extent, hence its transmissivity and storage/recharge potential). Surficial aquifers are widely distributed throughout the Prairies. They range

in size from small, highly localized zones, to extensive buried valley systems hundreds of kilometres long. For instance, the Hatfield Valley Aquifer, a buried pre-glacial valley, stretches in a southeasterly direction from Cold Lake in the west to Langenburg near the Manitoba border. A major intertill aquifer within Saskatchewan is the Regina Aquifer, which is associated with the Condie Moraine extending north from Wascana Creek. The yield of wells in the Hatfield Valley and Regina Aquifers is in the range of 5-55 L/s (MLM Ground Water Engineering, 1982). Lissey (1962) found TDS concentrations of 2000 mg/L in the Regina Aquifer. For the Hatfield Aquifer, and other buried pre-glacial valleys within the region, TDS values ranged from less than 500 to greater than 5000 mg/L (MLM Ground Water Engineering, 1982).

The Assiniboine Delta Aquifer, located in the Brandon area, is the largest surficial aquifer in Manitoba. Of the total estimated potential land under irrigation in Manitoba, the Assiniboine Delta Aquifer accounts for approximately 22 per cent of that supply (Swift, pers. comm. 1982). Yields from wells in the area range from 25 to 70 L/s with a total sustainable discharge from the aquifer (without "*mining*" the ground water) amounting to 2000 L/s (SNBB, 1972). The quality of ground water produced by this aquifer is good, with TDS values ranging from 200 to 500 mg/L.

Bedrock Aquifers

Beneath the agriculturally significant parts of the Prairie Provinces, bedrock ranges in age from Tertiary to Precambrian. The sedimentary sequence lying upon the Precambrian basement occupies the Western Canada Sedimentary Basin. Usable ground water resources are available above some defined "*base of exploration*," below which ground waters are either too saline, or are simply uneconomical to extract. In Alberta and Saskatchewan, the base of exploration is at the upper-middle Cretaceous contact. In Manitoba, it is at the Paleozoic-Precambrian unconformity. The yield of bedrock aquifers and the ground water quality vary considerably from place to place.

One of the most heavily used bedrock aquifers in western Canada is the Carbonate Aquifer in Manitoba. It is capable of providing large yields (up to 40 L/s) of water with a TDS concentration less than 1000 mg/L. It supplies approximately 17 per cent of the City of Winnipeg's water (Render, 1970). Bedrock aquifers in southern Saskatchewan are used to a limited extent and include the Wynyard, Ravenscrag, Frenchman, Whitemud and Eastend Aquifers. Water quality is variable throughout these aquifers, with TDS concentrations ranging from 250 to 4000 mg/L. Anticipated well yields are generally in the order of 0.5 L/s (MLM Ground-Water Engineering, 1982). Southern Alberta contains two main bedrock aquifers, the Milk River and the Old Man. The Milk River Aquifer in places may yield up to 40 L/s. However, water quality is poor in its northern sections where TDS concentrations approach 4000 mg/L. The Old Man Aquifer, located in the southeastern portion of Alberta, is generally of poor quality with low anticipated yields. Hence it is unsuitable for most irrigation purposes.

It can be seen from the preceding discussion that quality constraints impose severe limitations on what can be termed an aquifer for purposes of irrigation in western Canada. Some of the surficial aquifers, most notably the Assiniboine Delta Aquifer in Manitoba, are suitable in both quality and quantity. Surficial aquifers generally contain the best water because they are uncon-

fined and thus able to receive recharge very quickly. Unfortunately, these characteristics also make them particularly vulnerable to contamination from irrigation return flow.

2.5.2 Soils

Soil texture and fertility are two additional factors that influence the location of irrigation projects and their impact on ground water quality. Soil texture influences the rate of infiltration and, together with fertility, is a determining factor for crop location as well as the location of irrigation.

Most of the soils within the agricultural areas of western Canada are of the Chernozemic Order, which includes the Brown, Dark Brown, Black and Dark Grey groups (Leahey, 1969). Figure 2.2 shows the approximate distribution of these groups. Also important, but to a lesser extent, are the Brown Solonetzic group found in western Saskatchewan. Luvisols are common within the region (Figure 2.2), but they generally do not coincide with irrigated areas which are the concern of this report. Most irrigation occurs within the Brown and Dark Brown soil groups. The remainder of this section relies mainly upon information contained in Soils of Canada, Agriculture Canada, 1977.

Chernozemic Soils

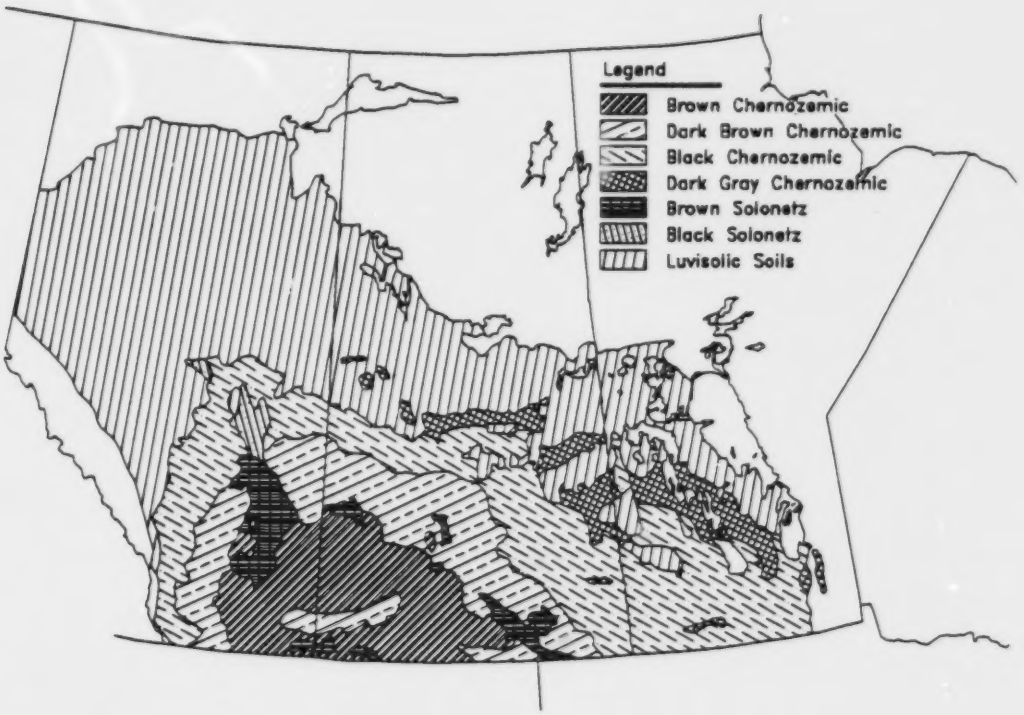
Brown soils are generally somewhat arid soils on the open prairie, and are predominantly loamy in texture with some sands and clays. Less than 50 per cent of these soils are cultivated. Cultivation is largely restricted to the finer textured loams or clays which are capable of retaining moisture. Coarser textured areas require irrigation in order to make crop production feasible. The fertility of Brown soils is reasonably good, with the primary limiting factor being moisture deficit. Some of the finer textured soils which are irrigated often have problems related to adequate drainage and salinity.

Dark Brown soils are somewhat arid soils found mainly in the open prairies of Alberta and Saskatchewan. They are predominantly loamy in texture, although there are significant Dark Brown soils of a sandy or clay texture. Approximately 70 per cent of these soils are cultivated, with the coarse textured areas being used for pastureland. Since the fertility is adequate, the major limitation to productivity is moisture availability.

Soils of the Black group occur in the better moisture conditions of the Parkland belt (Saskatchewan Agriculture, 1988). They are primarily loamy, but may also be of a coarse sandy, or fine clay texture. Limitations to the fertility of Black soils may be encountered where the soils are highly calcareous, or have a coarse, sandy texture.

Dark Grey soils are mostly loamy and occur primarily in areas of rolling topography, in the parkland-forest transition area. They are highly productive with occasional moderate limitations due to permeability. Due to limitations imposed by topography, irrigation is usually minor in areas where these soils are found.

Figure 2.2 : Common Soil Groups of Agricultural Areas of the Prairie Provinces.



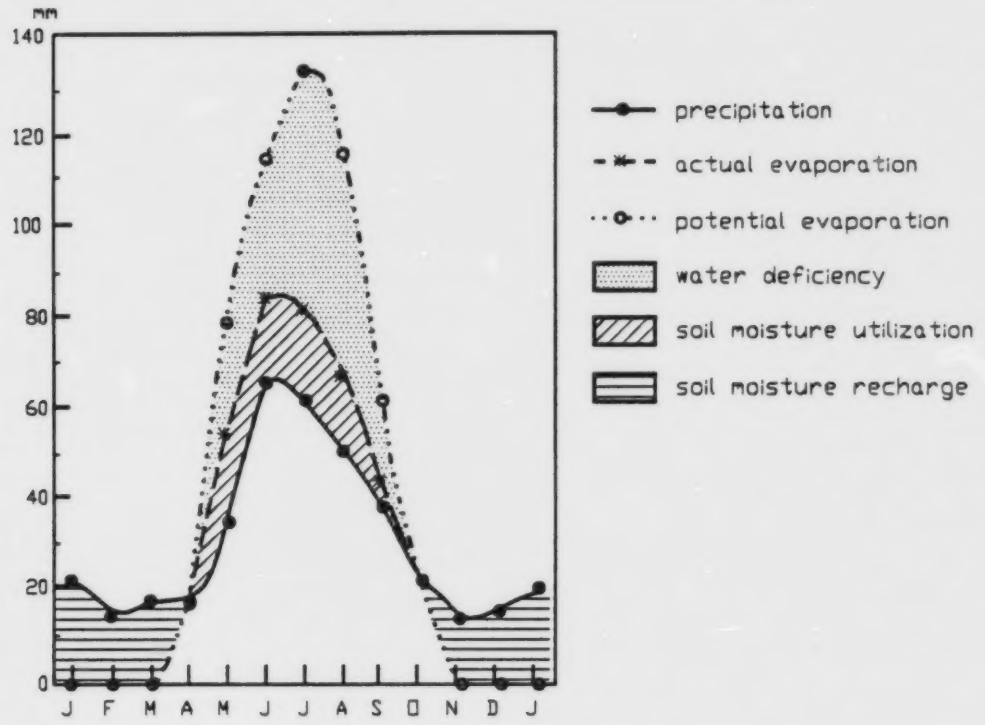
Solonetzic Soils

Solonetzic soils are formed from the salinization of parent materials by alkaline salts followed by, or concurrently with, desalinization through processes of leaching. Structural limitations of this soil type relate to its plasticity when wet and hardness when dry. This restricts moisture penetration and hence crop growth. Periodic salinization also poses a problem which can often be accelerated by irrigation.

2.5.3 Climate

The Prairie Provinces are characterized by relatively low precipitation and extremes of temperature from over 38 °C to less than -45 °C. The average annual precipitation ranges from 28 to 51 cm. Moisture requirements are often not met, and water deficits (where evapotranspiration exceeds annual precipitation) are experienced in many areas (Figure 3.1).

Figure 2.3: Water Balance for Saskatoon, Saskatchewan



The coldest month is January, with mean daily minimum temperatures ranging from -12/C to -31/C. In the spring, the area between Medicine Hat and Lethbridge in southern Alberta is the warmest region in western Canada, with the growing season beginning around mid-April (threshold temperature of 5.6/C). The growing season begins in most of the remaining agricultural areas in late April, although the risk of frost continues in most areas until about June 15. The area with the longest frost-free period (140 days) is found in southeastern Alberta and southwestern Saskatchewan.

The Prairie Provinces also experience large diurnal variations in temperature. Variations in topography and the absence of bodies of water are the primary reasons. Areas with low humidity, such as the Prairies, have a greater daily range of temperatures than those with high humidity levels.

Approximately 50 to 70 per cent of the annual precipitation falls between May and September. The snow depth over most of the agricultural regions is less than 0.3 m at the end of February, and by the end of March very few agricultural areas have more than 5 cm. Precipitation levels begin to rise in May. High surface temperatures cause showers and an increasing number of thunderstorms toward mid-summer, with the maximum monthly precipitation falling in June or July. Thunderstorms most frequently occur in July.

These tend to be highly localized and very short in duration, generating intense rainfall. After July the number of thunderstorms decreases rapidly.

Maximum soil moisture utilization coincides with the maximum precipitation period in western Canada (Figure 2.3). The Prairies are usually water-deficient however, and the actual evapotranspiration from the surface will generally be markedly less than the mean lake evaporation (Ferguson, O'Neill, and Cork, 1970). It follows, that potential evapotranspiration will exceed actual evapotranspiration during the hottest part of the growing season, as soil moisture has become completely utilized and the actual volume of precipitation is not sufficient to make up this shortfall. A deficit in terms of seasonal water balance results. Figure 2.3 illustrates this for the area of Saskatoon.

3. IRRIGATION AND GROUND WATER CONTAMINATION

3.1 Salinity and Ground Water

3.1.1 Overview

Salinity problems have been associated with irrigation practices for many centuries (Pillsbury, 1978), and remain a major issue of concern for irrigated lands (Jame, pers. comm., 1982). The problem is in the build-up of salts which affect crop yields. Soluble salts interfere with the ability of plants to absorb water. These include the sulfates, chlorides and carbonates of magnesium, calcium and sodium (Holm, n.d.).

All irrigation water contains salts, as no natural source of water is completely salt-free (Pillsbury, 1978). When irrigation water is applied to a field, salts tend to accumulate in the upper soil profile as water evaporates or is transpired by plants. The rate at which salts accumulate in soil is dependent on many factors, including the quantity and quality of water applied, soil type, and climate. In order to ensure the longevity of irrigation projects, soils must be protected against the accumulation of excess salts. This is accomplished be the process of leaching. Crucial to this is the determination of the correct degree of leaching, as too little will bring about the accumulation of salts in the soil zone, and too much will remove nutrients, and waste irrigation water. Generally, the surface component of irrigation return flow contains only a slightly higher concentration of salts than the water applied (Skogerboe and Walker, 1981). The water percolating through the profile will contain most of the salt left behind by the water as it was lost through evaporation and transpiration.

3.1.2 Leaching Fraction

To attain adequate leaching, more water is applied than is needed by the crop to cause a portion of the applied water to percolate through the root zone. The amount of water intentionally leached through the root zone is called the leaching fraction (LF) (Jame and Nicholaichuk, 1981). As subsurface water moves through soil it may dissolve the salts present in the soil and eventually transport these with its original salt load to ground water (Rude, 1982). With a high LF, the soil salinity profile tends to be uniform. The salinity profile with a low LF varies a great deal, with the highest salt concentrations accumulating in the root zone (Bachman *et al.*, 1980). In the 1950's the United States Department of Agriculture (U.S.D.A.) developed LF recommendations for agricultural crops to ensure acceptable soil conditions on irrigated lands. A LF of 25 per cent (0.25) was recommended for crops with a moderate salt tolerance, such as alfalfa. This resulted in large quantities of water being applied with much percolating through the soil on a routine basis. In recent years it has been determined that this fraction was higher than was necessary to maintain an acceptable soil salinity profile and this recommendation has been altered to prohibit excessive use of water (Jame, pers. comm., 1982).

Jame and Nicholaichuk (1981) have recently done a comparative study of salinity in soils with different leaching fractions. Crop yields and water consumption were measured using a LF of 10 per cent (0.1) and 25 per cent (0.25). It was found that the reductions in crop yield associated with slightly increased soil salinity under the low LF was compensated through a significant increase in water available for irrigation of more land. With a LF of 0.1, 1.3 hectares could be irrigated with the same amount of water used for 1.0 hectares at a LF of 0.25. Most significant is the fact that the quantity of drainage water was only 750 m^3 with the low LF compared with 1700 m^3 for high leaching conditions. Because significantly greater amounts of water moved through the soil under the larger LF, more salt could move out of the root zone toward ground water.

Many models of varying complexity have been developed to predict salt concentrations, movement and emissions below irrigated fields. For a discussion of some of the models, the reader is referred to Jury and Pratt (1980), Suarez and Van Genuchten (1981), and Skogerboe and Walker (1981).

Rhoades and Suarez (1977) cited studies which showed that the salt load moving from the root zone can be lowered by 2 to 12 tonnes/ha/year by reducing the LF from 0.3 to 0.1. They suggested three primary explanations for this. First, with a reduced LF, less salt is discharged because less water is applied. Secondly more salts precipitate under a lower LF. The calcium (Ca), bicarbonate (HCO_3) and sulphate (SO_4) ions in the applied water will precipitate as gypsum minerals and carbonates. Lastly, fewer additional salts are picked up from the dissolution of minerals in the soil. Rhoades and Suarez (1977) stated, "*the preceding analyses and data clearly demonstrate that decreasing the leaching fraction can significantly decrease the salt burden of drainage waters from root zones.*" A study by Rhoades, *et. al.* (1974) examined the effect that leaching fractions had on the dissolution and weathering of soil minerals and salt precipitation in the soil. They found that as the LF decreases, the precipitation of salts in soil increases and there is less weathering and dissolution of salts. Therefore, it is clear that reduced leaching will almost always reduce the salt load discharged in the drainage waters. It is not always the case, however, that a lower LF will lead to less salinization of ground water. This is further discussed below.

3.1.3 Drainage

In order to maintain a sufficiently high LF, soils must have sufficient drainage to allow water to move downward, away from the root zone where salt accumulation can occur. If drainage is restricted, excessive applications of water combined with seepage losses from canals can cause the ground water level to rise (Jones and Lebedin, 1986). As the water table rises, it carries salts toward the soil surface. Capillary action may then result in the upward movement of water to the surface if the water table is high enough. After the water evaporates, the salts are left behind on the ground surface. Skogerboe and Walker (1981) stated, "*this process of salinization has not only resulted in declining agricultural production, but has caused many lands to become essentially barren.*"

A related salinity problem which may affect dryland farming areas as well as irrigated area is that of the saline seep (Duaime *et al.*, 1986). Saline seeps

are formed when percolating water encounters an impermeable layer and flows along that layer laterally until reaching a ground water discharge area. The seep water causes the water table to rise and the water reaches the surface, evaporating and leaving a saline patch (Holm, n.d.). In North Dakota, the progressive deterioration of ground water quality has been related to elevation, salinity, and permanence of surface water bodies along a localized ground water flow system (Arndt and Richardson, 1986). This demonstrates how the susceptibility of the soils of a given area to salinization, relates to existing ground water flow, and the positioning of that area above the ground water flow system.

3.1.4 Salinity of Ground Water

It has been established that ground water quality will be affected by the amount of leaching, and hence upon drainage practices. Furthermore, the overall cumulative impact that these practices will have on the ambient salinity levels of ground water will depend upon the hydrologic characteristics of the particular ground water system.

If a ground water flow system is hydrologically isolated, and the concentration of salts in the leached water exceeds that of the ambient ground water, then the concentration of salts in the ground water will eventually approach the salt concentration of the subsurface drainage water (Rhoades and Suarez, 1977). This, however, may take many decades, depending on the leaching fractions being applied. Reduced leaching increases the drainage water's residence time in the upper soil profile considerably and thus causes a delay in ground water salinization. "The considerably slower travel times for low leaching means that the saline drainage front takes much longer to mix with and degrade the higher quality ground water" (Suarez and Van Genuchten, 1981).

A low LF means that a small amount of the applied water remains in the soil to percolate to the ground water. A high LF means that larger amounts of the applied water remain to leach through the soil. Given a fixed content of salts in the drainage water and soil, a lesser amount of water will be more concentrated with these salts than a greater amount of water with the same amount of salts present (i.e. dilution factor). This causes the drainage water for a low LF to become saturated with salts more quickly and accordingly cause any excess salt to precipitate in response to evapotranspiration. Suarez and Van Genuchten (1981) found this to be true using a theoretical model. They concluded that the greater the amount of precipitable salts in the applied water, the greater benefit reduced leaching will be to ground water quality.

In general, if the receiving ground water can accommodate more Cl, SO_4, and HCO_3 in solution, then drainage water with a higher concentration of salt that reaches the ground water will cause increased salinization of that water body. Rhoades and Suarez (1977) suggest that for a period of time a low LF will cause a lower salinization of ground water than a high LF in an aquifer whose only recharge source is the drainage return flow. However, this is only true until all the mixed ground water has been replaced by drainage water. After that time, a low LF will cause a greater increase in the salt concentration of ground water than a high LF.

When irrigation water is taken from outside an aquifer which is hydrologically connected to other water systems, the effect that different leaching fractions will have on the salinity of ground water will be determined largely by two factors: (1) the background salt concentration of the ground water; and (2) the potential for high quality recharge from a source other than the irrigation return flow. With regard to the various salt concentrations of the receiving ground water, the effects of different leaching fractions of varying water quality are analogous to the closed basin situation. If there is a sufficient quantity of good quality water also recharging the aquifer, reduced leaching will always reduce the salinization of ground water. Rhoades and Suarez (1977) stated " . . . *low leaching can continuously reduce degradation of the ground water, only if other sources of high-quality recharge into the basin exist and if flow out of the basin is adequate relative to the drainage inflow.*"

3.1.6 Summary

In summary, leaching of salts from the soil surface and upper root zone is necessary on irrigated land to prevent the build-up of salts to a point at which crop growth would be severely impaired. A high LF carries a greater salt load out of the root zone than a low LF. However, a large net transfer of salt to ground water may not degrade its quality as significantly as a low LF with a higher concentration of salt in the leachate over the long term. The impact that varying leaching fractions have on the quality of ground water depends on several major factors. One of these is the quality of the drainage water, which is dependent on the quality and quantity of applied water and the salts present in the soil. A second factor is the quality of the receiving ground water and its ability to accommodate more salts in solution. A third consideration is the prevailing hydrologic conditions which include recharge from all sources as well as discharge from the aquifer. Increased irrigation efficiency increases the travel times of the drainage water and retards the rate of ground water salinization. Depending on the quality of the drainage and ground water, this efficiency may also significantly lower the salinity of ground water over the long term (greater than 100 years) in ground water systems.

3.2 Effluent Irrigation and Ground Water

3.2.1 Overview

Sewage effluent has been used for agricultural purposes for centuries and in many parts of the globe, particularly in Asia. In addition to solving disposal problems, crop production can also be increased considerably. Effluent irrigation studies in the Prairie Provinces began in earnest only in the early 1970s (Nicholaichuk, 1981). A major stimulus for effluent irrigation comes from the fact that many urban centres are faced with waste disposal problems and need to find some means of safely disposing of municipal wastes (Wiebe, 1982). At present there are fourteen effluent irrigation projects in Saskatchewan with a total of 1,146 hectares under irrigation (Wiebe, 1982).

Alberta has twenty-two projects, and Manitoba has only one project (Nicholaichuk, 1981).

In western Canada, there are two basic crop effluent irrigation methods — slow rate and high rate irrigation (Lawson, 1979). The objective of the first technique is to have most of the effluent taken up by the plant while meeting the minimal leaching requirements. As a result only 10-20 per cent of the irrigation water will reach the ground water. High rate irrigation utilizes much higher loading rates, (1.0 to 6.0 metres annually compared with 0.3 to 1.0 metres annually for slow rate crop irrigation), which result in greatly increased quantities of applied water and pollutants reaching the ground water (Lawson, 1979). Lawson (1979) suggested that the amount of water percolating to the ground water annually may vary from 0.1 to 0.3 metres under low rate irrigation, and from 0.3 to 5.0 metres under high rate application. Oldham (1976) suggested that the greater the amount of water applied, the greater the potential for ground water contamination. The major contaminants related to effluent irrigation are phosphorus, heavy metals, bacterial contaminants, nitrogen, and salts.

3.2.2 Phosphorus

Phosphorus is a significant component of sewage effluent. It would appear that phosphates do not generally pose a serious threat to ground water. Oldham (1976) noted that phosphorus (P) is highly adsorbed in the soil profile and is generally not mobilized by biological action. Lawson (1979) concurred: "*A large number of studies have shown that phosphates are largely adsorbed into the soil structure and do not percolate down into the ground water.*" In a soil column study of land irrigated with municipal effluent, Penkava and Ray (1982) found no significant changes in P concentrations in the saturation extracts, despite varied loading rates. A five year study investigating the distribution of P in silt loam soil irrigated with municipal waste water found P to a depth of 60 cm (Latterell *et al.*, 1982). The high rate applications averaged 236 cm of effluent applied annually with an average PO_4-P concentration of 6.9 mg/L. They concluded that this was the maximum rate that could be applied without degrading the ground water which was 140-150 cm below the surface. The increase in downward mobility of phosphate after the saturation point has been reached could be identified and specified through further research.

3.2.3 Heavy Metals

Heavy metals which may be present in municipal sewage normally accumulate in the sludges isolated in waste treatment (Lawson, 1979). The heavy metal concentrations in liquid effluent tend to be much lower and thus the land application of sludges pose the greatest potential for contamination. Any toxic metals that are applied to land tend to concentrate near the soil surface with very little leaching (Curran, 1981). After more than eighty years of effluent irrigation near a major European city, heavy metal contents were detected in the upper soil layers which surpassed the tolerance limits of the country (Milde *et al.*, 1981). The study concluded, "*exposure to sewage over many years will result in contamination by heavy metals which, however, remain essentially limited to the upper soils*" (Milde *et al.*, 1981). Thus it

appears that any heavy metals present in the effluent used for irrigation will not pose a significant threat to ground water.

3.2.4 Nitrogen (Nitrates)

Nitrogen (N) in sewage effluent is largely in the form of unionized ammonia (NH_3). Ammonia may be adsorbed by soils, but over time will convert to nitrate (NO_3) under aerobic conditions. Ammonia is relatively insoluble in water, while nitrate is very soluble. If not utilized by the crop, nitrate easily percolates through the soil profile under excessive moisture conditions. "*The combination of N added in effluent used for irrigation, and the necessity of a leaching environment may result in substantial loss of N by leaching of NO_3-N*" (Lund *et al.*, 1981). If nitrate leaching through the soil encounters an anaerobic zone, denitrification could take place. The nitrate may eventually be reduced to N_2 (gas) and removed from the percolating water. Nitrogen speciation in ground water has been shown to be strongly influenced by redox conditions (Hendry, McCready, and Gould, 1984). Denitrification occurs under conditions of low oxygen potential with the transformation of nitrate (NO_3) to ammonium ion.

Lund *et al.*, (1981) referred to studies where 19 to 50 per cent of the N added from the effluent irrigation leached below the root zone. In his study of loamy sand soil that had been irrigated for more than ten years, the mean NO_3-N concentrations in the soil solutions, found below the root zone (1.8 - 6.0 m depth), ranged from 15.4 to 32.4 mg/L. The average total N concentration in the applied effluent was 23.0 m/L. The water application rate and the coarse-textured soil led to a high leaching fraction. The study concluded that, " . . . *the large leaching fraction leads to substantial mass emissions of NO_3*" (Lund *et al.*, 1981). After taking into account crop removal of N, volatilization and denitrification, leachate concentrations accounted for 51 to 76 per cent of the applied N. This indicates that the amounts of N that are leached from an area irrigated with effluent can be substantial.

In a study of the water quality around the effluent irrigation project at Taber, Alberta, three of the four wells placed along the length of the irrigated area showed an increase in NO_3 concentration over a three-year period, although no definite trend was established for the NO_3 concentration in the ten wells tested (Olynyck and Graveland, 1974). Chopp *et al.*, (1982) in a study of ammonia oxidizing bacteria in soils irrigated with municipal waste water, found that peak NO_3 concentrations in ground water (water table at 140 cm) occurred one month following the peak NO_3 concentrations in the soil. The mean total N concentration in the effluent was 22.6 mg/L, while the peak nitrate concentration in the ground water under high application rates (1.3 cm per hour for 8 hours a week) was 15.0 mg/L with a mean concentration of 11.1 mg/L over the 5-month sampling period. The peak concentration of NO_3-N in the ground water under low application rates (1.3 cm per hour for 4 hours a week), was 10.6 mg/L with a mean concentration of 5.5 mg/L over the 5-month period.

These studies substantiate the hypothesis that nitrogen contamination of ground water will occur with effluent irrigation under certain conditions. Lawson (1979) constructed a typical analysis for a nitrogen balance under slow rate crop irrigation (Table 3.1). The amount of N entering ground water

was calculated to be 15 per cent of the total nitrogen loading at the soil surface. The concentration of N in the percolate was calculated to be 6 mg/L compared with a N concentration of 25 mg/L in the effluent. Crop uptake accounted for 75 per cent of the total N load and denitrification accounted for 10 per cent.

Table 3.1: Nitrogen Balance for Slow Rate Irrigation

Wastewater loading (metres/year)	1.6
N concentration in Effluent (mg/L)	25
N concentration in Percolate (mg/L)	6
N loading (kg/ha)	400
N to ground water (kg/ha)	60
Crop Uptake (kg/ha)	300
Denitrification (kg/ha)	40

Source: adapted from Lawson, 1979, Table 2

The five major factors affecting the potential for N to reach ground water are: (1) the concentration of nitrogen in all its forms in the sewage; (2) the application rate; (3) the nitrogen balance for the crops irrigated; (4) the soil type and chemical characteristics; and (5) the level of microbial activity in the soil. High rates of effluent application on to coarse-textured soils will favour the entrance of nitrate into ground water. Effluent with high N concentrations that is applied in excess of crop N requirements could also result in nitrate contamination of ground water. Nitrate is discussed further in 3.4 in relation to fertilizer losses.

3.2.5 Biological Contamination

Another major concern with sewage irrigation is the potential health hazards associated with the transmission of bacteria and viruses to ground water. Continuously moist conditions and low temperatures in soil substrates favour the survival of most bacteria which can move great distances in very coarse soils (Keswick and Gerba, 1980; Curran, 1981). Under these conditions bacteria and viruses may persist many months. Although viruses can be adsorbed to the soil (Oldham, 1976), they may still remain a threat to ground water. Lawson (1979) stated, " . . .*it appears they (viruses) may survive for long periods of time, up to three months, and may be eluted by subsequent application of effluent, percolating down to the ground water table.*" The pH of the unsaturated zone is an important factor affecting the adsorption of viruses. In an alkaline environment there is reduced binding of viruses to the soil which favours increased movement (Curran, 1981). Diffusion processes are insignificant and the contaminants move slowly with the ground water itself (Lawson, 1979).

It appears that, with the exception of particularly coarse soils, most biological organisms are filtered out in the upper soil layers. Biederbeck (1977) reported that profiles of clay-loam soil irrigated with effluent showed that over 90 per cent of the fecal bacteria were retained in the top 2.5 cm of soil. No movement of fecal bacteria occurred below 0.3 m Curran (1981) states that over 95 per cent of the enteric bacteria in effluent can be retained in the top 2.5 cm of a clay loam soil under slow rates of irrigation. Milde *et al.* (1981) found that coliform bacteria were reduced by 99 per cent to depths of 0.3 m. Neither coliforms nor Salmonella or Shigella species were found below 0.3 m.

Sewage treatment can greatly reduce the number of bacteria and viruses in the effluent, thereby reducing the threat of biological contamination of ground water.

The die-off of bacteria also increases with intermittent wetting and drying of the soil (Lawson, 1979). In addition, sunlight rapidly kills any bacteria present at the soil surface (Biederbeck, 1982). Thus, if the soil is effective in filtering out any bacteria that are present in the effluent, climatic factors act upon these bacteria to eliminate them.

3.2.6 Salts

Salinity is a major consideration with effluent irrigation. For instance, the main problem encountered by users of effluent from Swift Current is the high salt concentrations of that source (Nicholaichuk, 1981). This requires additional effluent in order to leach salts out of the root zone. Bole *et al.*, (1981) concluded from their study of waste water irrigation, that "*the main concern under this waste water irrigation regime seems to be in keeping the LF at 15 per cent to prevent a large net transfer of salt in the ground water system...*"

In the Taber effluent irrigation project (Section 3.2.5), the two parameter changes considered significant were a rise in the water table and " . . .*a considerable increase in chloride concentration in the ground water*" (Olynyck and Graveland, 1974). In the study by Lund *et al.* (1981) referred to earlier, the chloride concentration measured in the soil solution below the root zone (1.8 to 6 m) ranged from 9.5 to 13.4 meq/L (332 to 470 mg/L), while the concentration of chloride in the effluent being applied was 7.9 meq/L (270 mg/L). Shelton *et al.* (1981) examined the concentrations of selected constituents in surface runoff and ground water in a watershed after municipal sewage sludge was applied to the land (silt loam soils). He found a significant increase in sodium concentration of the ground water and attributed it to the leaching of salts. Penkava and Ray (1982) found " . . .*a substantial amount of sulphates*" in the (subsurface) drainage water after a 110-day simulated growing season on loam and silt-loam soils. They suggested that the expected increase in soil salinity at the end of the growing season would be mitigated by the leaching of the accumulated sulphates out of the root zone (and eventually into the ground water) by winter precipitation and precipitation in wet years.

Ground water of good quality will exhibit measurable increases in salt concentrations as a result of long term effluent irrigation return flows. "*In considering the effects on the ground water it is important to remember that in many cases the impacts take place slowly and over a long period of time as compared with discharges to surface watercourses*" (Lawson, 1979). This

concern also applies to other potential contaminants such as pesticides, which will be discussed below.

Milde *et al.* (1981) discussed a study area located in Germany, of predominantly sandy loam soils which has been irrigated for over eight decades. The influence of sewage irrigation on ground water quality is noted by comparing the measured concentrations of various constituents in ground water exposed to sewage irrigation and ground water not influenced by irrigation (Table 3.2). In all cases, significant increases in concentration were observed for salinity (as measured by electrical conductance), ammonia, nitrate, phosphate, chloride and sodium. This seems to suggest that in the long term, measurable increases in these and possibly other contaminants will be found in ground water affected by effluent irrigation.

Table 3.2: Impact of Effluent Irrigation on Ground Water Quality

	Ground Water Exposed to Sewage	Unexposed Ground Water
EC (μS)	1500	90-788
NH_4 (mg/L)	0.2	0.01
NO_3 (mg/L)	280	0.01-95.0
PO_4 (mg/L)	23	—
Cl (mg/L)	168	8-61
Na (mg/L)	142	2.5-32.4

3.2.7 Summary

These studies suggest that, in the long term, effluent irrigation can have a significant effect on the chemical quality of ground water. The major factors affecting the impact of effluent irrigation are the chemical quality of the effluent, the rates of application, the ambient quality of the ground water, the soil type and the type of crop being grown. Nitrogen and salts appear to pose the greatest potential threat to the quality of ground water underlying effluent irrigated land.

3.3 Pesticides and Ground Water

3.3.1 Overview

Over the past decade there has been increasing concern about the threat of pesticides (insecticides and herbicides) to water systems. However, the potential for pesticides to contaminate surface water has been investigated more thoroughly than their potential to contaminate ground water. In spite of the lack of extensive study on this topic in Canada, pesticides have been recognized as a potential threat to ground water. "*Pesticides may be significant in agricultural areas as a diffuse source of ground water pollution*" (Everett, 1980). The primary factors affecting the downward movement of pesticides in soil and their potential for entering ground water are soil properties, pesticide solubility, mobility and persistence, and the soil moisture conditions. These variables must be considered when attempting to predict the potential for pesticides applied to irrigated land to enter ground water.

3.3.2 Soil Properties and Pesticide Movement

Fine textured soils (clays) have a high sorptive capacity (LeGrand, 1966). One reason for this is that clay particles have surfaces with unbalanced electrical charges and thus will attract charged particles such as ions. Organic matter also enhances sorption (Wescott, 1976, EPA-Office of Water Programs, 1972). Organic matter has a highly reactive surface area, and has a cation exchange capacity which is greater than clay (Bailey and White, 1970). Singhal and Bansal (1978) found that a destruction of organic matter in the soil reduced the adsorption of the pesticides onto the soil particles and increased their ability to move through the soil.

3.3.3 Solubility

The degree to which a pesticide is sorbed onto soil particles is dependent largely upon the pesticide's solubility, its vapour pressure, the molecular properties (size, polarity, functional groups), and the makeup of the soil particles (Tinsley, 1981). The partitioning of a pesticide between the interstitial water and the soil solids is described by an isothermal relationship, expressing the equilibrium between the concentration of a pesticide in the aqueous (soil moisture) phase, and the solid (sorbed) phase. Generally, lower water solubility in a compound implies easier adsorption.

A pesticide's vapour pressure also affects its solubility. The Henry's Law constant relates these two properties to one another. It is approximately the ratio of the pesticide's vapour pressure to its solubility in water, at the same temperature (Cohen *et al.*, 1984). The solubility of a pesticide greatly affects its movement through the soil profile. It has been shown that the greater the solubility of a pesticide, the higher is the fraction that will be lost through the water runoff (Wauchope, 1978). Adams (pers. comm. to Dr. J. Bauder; International Garrison Diversion Study Board, 1976) suggested that pesticides in concentrations over 100 mg/L have potential to be leached. Gener-

ally, the more soluble pesticides are more susceptible to downward percolation (Wauchope, 1978; Grover, 1976).

3.3.4 *Mobility*

The mobility of a pesticide within the soil column relative to other pesticides can be determined by comparing its solubility and sorption potential. A highly mobile pesticide would have to be highly soluble and resist sorption. Any pesticide with such characteristics would move easily and rapidly through soil. Pesticide mobility is also affected by the ambient quality of ground water in the unsaturated and saturated zones. The term "*facilitated transport*" has been coined to describe the process by which a compound which would normally be of a low mobility in relatively pure ground water, might have a much higher "*apparent*" mobility in a ground water with a naturally high dissolved organic carbon concentration. The natural organic compounds may compete with the pesticide for sorption sites on the subsurface particles, thus reducing the degree of sorption of the pesticide, and making it more mobile.

Mobilities may be expressed in terms of a range of classes as in Table 3.3, from immobile to highly mobile. Once in the ground water flow system, a highly mobile contaminant may be expected to migrate at a velocity of at least 50 per cent of the local ground water flow rate. A contaminant with a low mobility may move at less than 10 per cent of the ground water flow rate (Jackson, 1982).

Table 3.3: Mobility and Persistence of Common Herbicides

Herbicide	Mobility Class	Average Persistence
2,4-D	4	1
MCPA	4	1
Dicamba	5	2
Bromoxynil	4	1
TCA	5	1
Trifluralin	1	3
Triallate	2	1
Dalapon	5	1
Atrazine	3	4
Linuron	2	2

Class 1 - immobile	Persistence 1 - 0-2 months
Class 2 - slightly mobile	Persistence 2 - 2-6 months
Class 3 - slightly mobile	Persistence 3 - 6-12 months
Class 4 - mobile	Persistence 4 - 1-4 years
Class 5 - mobile	

Source: Adapted from International Garrison Diversion Study Board, Water Quality Report, Table A-21.

3.3.5 Persistence

The persistence of a pesticide (resistance to degradation) is also an important consideration for its potential entry into ground water supplies. Degradation of pesticides in ground water can take the form of chemical decomposition, volatilization, and microbiological degradation. Soil pH, leaching, and uptake by plants also affect the persistence of a pesticide in a given area. Wescott (1976) suggested that many pesticides have maximum stability at neutral pH levels, with variations from neutrality accelerating degradation.

Sorption influences the persistence of a pesticide by decreasing its availability for volatilization, microbial decomposition, and leaching. However, sorption may concentrate a pesticide in the upper soil layers where there is the highest microbial activity. This will contribute to its degradation providing that the micro-organisms are themselves not limited to attacking the pesticide in solution (EPA Office of Water Programs, 1972).

It has been suggested that the primary factor in the break-down of most pesticides is microbiological activity (Wescott, 1976; Weed Science Society of America, 1979). However, if it is true that microbial activity declines almost logarithmically with soil depth (Biederbeck, pers. comm., 1982) the length of time the pesticide remains in the top 15 cm will greatly affect its rate of breakdown (i.e. persistence time). The persistence of a pesticide will tend to increase as it moves downward through the soil. The findings of Smith *et al.* (1975) support this hypothesis. They found that the persistence of four herbicides under study increased with soil depth. The rapidity with which the pesticide moves downward determines the length of time it will remain available in order for microbial degradation to take place.

3.3.6 Moisture

The amount of moisture in the soil and the amount entering it at a given time are perhaps the most important factors affecting the downward movement of pesticides (Tinsley, 1979). In areas with low amounts of precipitation, only negligible amounts of pesticides might be expected to reach ground water (Grover, 1976). In areas where more moisture has been added than the root zone (about the top one metre of soil) can hold, some of the excess water will percolate through the soil transporting the pesticides in solution with it. Bailey and White (1970) referred to a study using tracers to investigate water movement through saturated and unsaturated soils that supports this assertion (assuming pesticides behaved similarly to the inorganic tracers).

The position of the water table also affects the likelihood of pesticides reaching ground water. Where water tables are high, the chances of ground water contamination are increased. Depending on drainage conditions and the amounts of water being applied to the land, the water table could rise under irrigated conditions. Grover (1976) noted, " . . . *in soils where water tables are high and rainfall (irrigation) moderate to heavy, significant amounts of herbicides may enter the ground waters.*" Bailey and White (1970) stated that the greater the amount and frequency of precipitation, the greater will be the depth and amount of pesticide leached in a given soil type.

An additional influencing factor is the rate at which the water percolates through the soil profile. Bailey and White (1970) suggested that increased rates of movement favour desorption of pesticides by influencing the equilibrium between the pesticide in solution and that adsorbed on to the soil colloids. The lower the solubility of a given pesticide, the less will be its tendency to desorb and be transported downward by soil water (MacKay and Yuen, 1980). Under low levels of moisture, there are proportionately fewer water molecules to compete for an adsorption site with the pesticide molecules than under high moisture levels where the water molecules effectively compete with the pesticide (Bailey and White, 1970). Soil texture greatly influences the rate of water movement, with the coarse textured soils allowing more water to pass through the soil at a greater rate than the finer textured soils. This is one reason why the fine textured clay soils in the Regina area, for instance, are not suitable for irrigation (Grover, pers. comm., 1982). Unfortunately, this implies that soil textures which favour irrigation also favour leaching of pesticides.

The results of a number of studies cited by Bailey and White (1970) seem to indicate that pesticides tend to leach to greater depths under lower intensities or increments of rainfall, given a constant total amount of water over a given time period. This indicates that under intense applications of water, more pesticides would be subject to surface runoff than to subsurface drainage, except where the soil in question is very highly permeable.

The timing of precipitation is also a factor in the potential leaching of pesticides. The longer the pesticide remains on the top soil layers, the more it will be subjected to the various processes of degradation mentioned earlier. Generally the longer the chemical is in the soil the harder it is to remove (Smith, pers. comm., 1982). If rainfall or irrigation occurs soon after chemical application, a pesticide will have a greater potential for reaching bodies of water through surface runoff and percolation before it is degraded to any great extent.

Other factors that may influence pesticide loss to the saturated ground water zone may include the method of irrigation and the method of pesticide application (e.g., an emulsion, with irrigation water, etc.). These factors may influence a pesticide's fate in the subsurface, and hence the behaviour and toxicity of the degradation products. More research needs to be done in this area.

3.3.7 Regional Application

Over 50 per cent of all crop land in the Prairies is treated with the herbicides 2,4-D and MCPA, while approximately 30 per cent of all crop land is treated with the wild oat herbicides triallate (Avadex BW) and trifluralin (Treflan) (Smith *et al.*, 1981). In Manitoba and Saskatchewan, two other herbicides widely used are Dicamba (Banvel) and Bromoxynil (Buctril) which are used on 28 and 25 per cent, respectively, of the area reportedly treated in the provinces (Table 3.4). Generic names of other herbicides used include Linuron, Atrazine, Roundup, Avadex, Stampede, Hoe-grass, Carbyne, Eradicane, Eptam, TCA, Avenge, Asulox, Dichloroprop and others (Daciw, 1981). The first seven herbicides on this list, as well as 2,4-D, MCPA, Bromoxynil and

Dicamba, are commonly used on irrigated land in Saskatchewan (Bill King, pers. comm., 1982).

Table 3.4: Herbicide Used on Cereal and Oilseed Crops

	Herbicide	% of reported area treated
Manitoba	2,4-D	21
	Trifluralin	18
	Triallate	16
	Bromoxynil	15
	MCPA	12
	Dicamba	10
Saskatchewan	2,4-D	41
	Triallate	22
	Dicamba	18
	Bromoxynil	10
	Trifluralin	10

Source: Apated from Smith, 1981.

In terms of the absolute amounts used, 2,4-D accounted for 46 per cent of the total; MCPA - 26 per cent; TCA - 5 per cent; Dalapon - 0.5 per cent; and all other herbicides together - 23 per cent (Table 3.5). This illustrates that while 2,4-D is used in the greatest amount, various other herbicides including Dicamba and Linuron together, are used almost as much (kg for kg) as MCPA (ie. 23 per cent vs. 26 per cent).

As indicated in Table 3.3 (Section 3.3.5), Dicamba has the highest leaching potential, with a mobility class of 5 and an average persistence of 2. This suggests it does not break down very fast, is not adsorbed and is very mobile (Smith, pers. comm., 1982). Dalapon has a high mobility and is leached readily in soils, but has a rather low persistence.

For many herbicides such as Dalapon and Dicamba, the degradation by soil micro-organisms accounts for the major loss (Weed Science Society of America (WSSA), 1979). Thus, the length of time that it remains in the upper soil layers is crucial to its persistence. TCA also leaches fairly readily in soil and is broken down by micro- organisms more slowly than Dalapon. 2,4-D, MCPA and Bromoxynil have mobilities which are less than the previously discussed three herbicides. 2,4-D and MCPA are broken down by microbiological action most quickly in warm moist soils. Atrazine does not have a very high mobility, but it is relatively persistent. Generally, Atrazine's leachability is limited by its adsorption to clay and organic soils, but desorption can also readily occur (WSSA, 1979). Pesticides with low mobilities but high persistence levels may still pose a threat to ground water. "*Even moving very slowly*

in the soil, a highly persistent pesticide may eventually reach underground water . . . " (EPA Office of Water Programs, 1972). Trifluralin and Triallate are relatively immobile, as they are strongly adsorbed by soil colloids (WSSA, 1979; Smith, pers. comm., 1982).

Table 3.5: Amounts of Herbicides Used in the Prairie Provinces

	2,4-D (000 kg)	MCPA (000 kg)	TCA (000 kg)
Manitoba	462.6	507.9	162.8
Saskatchewan	2318.2	644.1	149.1
Alberta	792.5	868.0	68.6
	Dalapon (000 kg)	Other* (000 kmg)	Wild Oat Herbicides** (hectares, treated)
Manitba	11.8	504.3	1,856,100
Saskatchewan	25.5	773.4	2,832,500
Alberta	0.3	536.0	2,443,900
* Bromoxynil, Dicamba, Dichloroprop, Linuron, Roundup, Stampede, Tordon 202C, etc.			
** Asulax, Atrazine, Avadex, Avenge, Carbyne/Wypout, Eptam, Eradicane, Hoe-grass, Mataven, Treflan			

Source: Apsted from Dacin, 1981.

Average persistence data must be used with caution as actual persistence can vary quite dramatically with temperature, soil pH, and soil-moisture ratio. In addition, most persistence data are related to pesticides in specific soil types, or in some cases distilled or natural surface water and not ground water (see for example EPA Office of Research and Development, 1976).

Of the herbicides most commonly used in the Prairies, Dicamba is regarded as having the greatest potential for entering ground water in irrigated lands (Grover, pers. comm., 1982; Smith, pers. comm., 1982). This is due to its high water solubility, resistance to adsorption, and relative persistence. In addition, because it is used extensively both on irrigated and non-irrigated land, it could have a significant impact on localized ground water quality. Its concentration and location in the soil profile is determined by the amount and frequency of precipitation and the original dosage applied (WSSA, 1979). Therefore, under heavy irrigation applications on sandy soil, with a high water table, Dicamba could pose a threat to ground water quality.

3.3.8 Treated Irrigation Ditches

One of the major uses of herbicides in western Canada is for the control of weed growth in irrigation ditches. Weed growth in intermittently filled irrigation ditches can impede the flow of water and reduce the carrying capacity of irrigation channels. Various herbicides such as Bromacil, Simazine, Monuron and Atrazine have been used to attempt to keep irrigation ditches free of weeds (Grover *et al.*, 1980). If these herbicides are applied in amounts necessary to control weed growth, it is possible that significant amounts may percolate through the soil beneath the ditches and enter ground water.

The movement and persistence of the four herbicides mentioned above was studied over a three-year period for intermittently filled irrigation ditches (Smith *et al.*, 1975). Three years after application, residues of Simazine, Monuron and Atrazine were recovered from all the soil depths sampled (0.0 - 90.0 cm). The highest concentrations of Monuron and Atrazine were found at the 45.0 - 90.0 cm depth interval while the highest concentration of Simazine was recorded at the 0.0 - 7.5 cm depth. Bromacil was almost completely absent from the soil profile to 90 cm. It was calculated that 15 per cent of the Monuron, 30 per cent of the Atrazine and 50 per cent of the Simazine still remained in the soil profile (to 90 cm) three years after application (these percentages were based on the assumption that the herbicides did not leach to depths greater than 90 cm). It was also found that over the study period, the concentrations of the four herbicide residues present in water in the ditches was inversely proportional to the water solubility of the herbicide. This meant that Simazine, which had the lowest solubility, was found in the highest concentration. Based on what is known about the downward movement of pesticides, it is possible to conclude that the most soluble herbicides leached to the greatest extent, leaving lower concentrations of these in the surface waters. Smith *et al.* (1975) made two general conclusions: (1) under conditions of irrigation, the herbicides Monuron, Bromacil and the triazines may leach to a depth of 90 cm or more; and (2) all the herbicides studied have long-term persistence (more than one year).

A two-year study (Grover *et al.*, 1980) followed the research of Smith *et al.* (1975) using the same ditches. The same four herbicides were reapplied and some foliar herbicides were applied for the first time. It was observed over time that the least mobile herbicide (Simazine) was largely contained in the upper 22.5 cm of soil while the most mobile herbicide (Bromacil) was concentrated in the 67.5 - 120 cm depth. The report concluded that, " . . . *maximum hazards to ground water contamination can be expected from the most mobile of the four sterilants, i.e., Bromacil*" (Grover *et al.*, 1980).

In a study of Bromacil in sandy soil (Hebb & Wheeler, 1978), it was found that under high rates of application, Bromacil was found in ground water (6 m depth) three months after application. The highest concentration (1.25 mg/L) was found in ground water 135 days after application. After one year, the concentration of Bromacil in ground water declined to 0.1 mg/L. Two years after application, only a barely detectable trace was found. In addition, it was discovered that peaks in measured residue concentrations were preceded one week by rainfall.

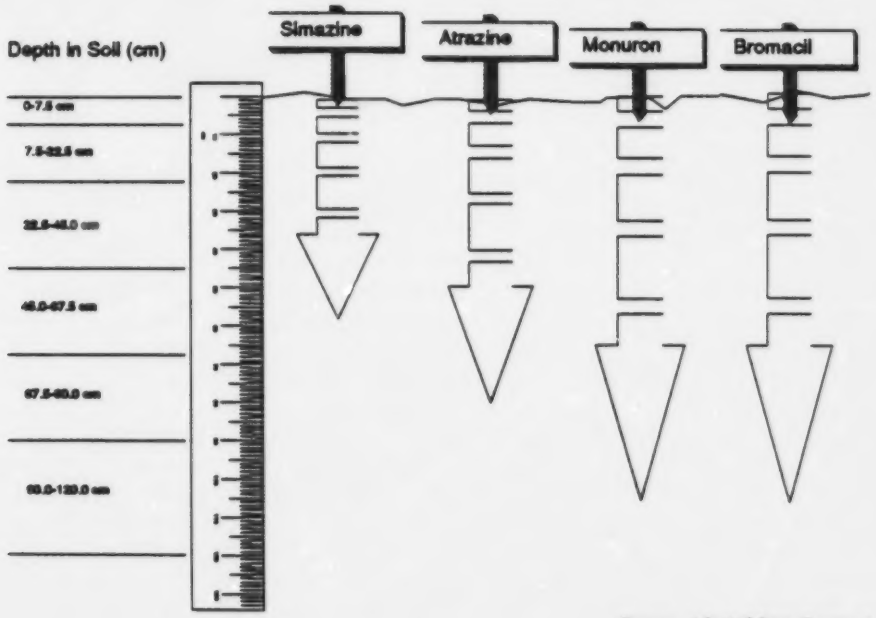
From the results of these three studies, combined with the knowledge of the herbicides' adsorption and leaching characteristics in basic soil types (WSSA, 1979), it appears that Simazine poses the least threat to ground water due to its low water solubility and high adsorption to soil constituents. Most of the Simazine applied will tend to remain in the top 5 cm of soil (WSSA, 1979). Atrazine has a higher water solubility than Simazine and is also quite persistent. Because of its persistence and lower adsorption compared to Simazine, Atrazine may have the potential to enter ground water over time. Monuron is relatively soluble and leaches rapidly from sandy soils. The fact that the highest concentrations were found in the 45-90 cm depth in the two year study (Grover *et al.*, 1980) indicates its high leaching potential. Bromacil is far more soluble than the other herbicides and has very low adsorption characteristics.

Table 3.6: Amount (%) of Herbicides Remaining After Three Years

Herbicide	Solubility (mg/L)	% of original amount remaining in 0-90 cm
Simazine	5	50
Atrazine	33	30
Monuron	230	15
Bormacil	820	negligible

Source: Adapted from Smith et al., 1975.

Figure 3.1: Leaching of Herbicides After Two Year



Source: Adapted from Grover et al., 1980

The relative depths at which the highest concentrations of the herbicides were found after the two-year study, and the percentage of the original concentrations found absorbed to the soil after the three year study, reflect their relative solubilities (Table 3.6, Figure 3.1). This further supports the theory that the most soluble pesticides will leach to the greatest depths and thereby pose the greatest threat to ground water.

3.3.9 Summary

Coarse soils with minimal surface area are not conducive to sorption. Soils with a low clay or low organic matter content also have low sorptive capacities, leading to few restrictions on the movement of pesticides through the soil. Soils with cracks, root-holes or other preferential passageways will likewise transmit water more rapidly downward, with attendant pesticide transport.

A pesticide's mobility, or ability to move through soils is a critical factor affecting its potential to enter ground water. Mobility is chiefly a function of solubility and resistance to sorption. The persistence of a pesticide also influences its ability to eventually reach the ground water table.

The timing, rate and frequency of water applications may also influence the leaching of pesticides into ground water. Saturated conditions favour desorption and increase pesticide mobility. Irrigating soon after chemical applications will also increase the likelihood that a pesticide will enter ground water.

Treated irrigation ditches present a unique situation where high localized applications of herbicide are combined with large seepage losses of water. Significant amounts of herbicide may be leached from these ditches and percolate to the ground water. Studies which monitor herbicide movement through the soil and herbicide concentrations in the ground water as well as the leachate are required to quantify and confirm this predicted phenomenon.

3.4 Fertilizer Nitrogen and Ground Water

3.4.1 Overview

The use of nitrogen fertilizer in North America has risen dramatically and steadily over the last three decades. On the Canadian Prairies in 1979, the use of nitrogen fertilizer was approximately 5.8 times that used in 1970 (Beaton, 1980). In 1978 approximately 30 per cent of the nitrogen fertilizer used in the prairies was in the form of urea, while ammonium nitrate captured 29 per cent of the market and anhydrous ammonia, 27 per cent. Urea-ammonium nitrate solutions (URAN) and aqua ammonia together accounted for 8 per cent of the market, and ammonium sulphate the remaining 6 per cent (Beaton, 1980). Because of this dramatic increase, the potential contribution of nitrogen to the deterioration of ground water quality is significant. This subject has received much attention in the United States,

Europe and elsewhere, but has not been studied to any extent in western Canada.

"*Nitrate, the oxidized form of nitrogen, is produced and accumulated in the soil regardless of the applied nitrogen form*" (Miller, 1977). Nitrate (NO_3) is water soluble and is generally unaffected by soil (Thomas, 1970). Because of this, it is a "*conservative*" species, and can migrate freely with percolating water. Therefore, the amount of water that moves through the root zone is a key determinant of the amount of nitrate that will eventually reach ground water. Robbins and Carter (1980) state that the amount of nitrate that will reach the water table is primarily a function of the water flux and the concentration of nitrate in the soil. Owen (n.d) found that a bromide tracer applied in similar quantities to nitrate on unirrigated pastureland appeared in peak quantities 21 to 26 months later in springs supplied by ground water flow from the test area. The implications for ground water quality are that if nitrate is applied in excess to plant needs, there can be a multi-year influence on ground water quality, even from a single application (Senft and Solomon, 1986). This indicates that, with applications of nitrate being equal, irrigation farming will cause a greater amount of nitrogen pollution of ground water than dryland farming.

3.4.2 Nitrogen Loss

The main pathways of loss of applied nitrogen fertilizer are crop uptake, volatilization, denitrification and leaching. It appears that crop uptake generally accounts for an average of no more than 50 per cent of the applied nitrogen (Saffigna and Keeney, 1977; Oakes and Young, 1981; Bouwer *et al.*, 1981; Keeny, 1982). Some data from the Big Spring River Basin in Iowa show a strong relationship between agricultural inputs and ground water quality. Hallberg (1987) shows that nitrate concentrations in ground water throughout this basin went from less than 1.0 mg/L in the thirties, to about 3 mg/L in the fifties and sixties, to an annual average of 9 mg/L in 1982, to 10.1 mg/L in 1983. Over the time of monitoring, the amount of nitrate applied as manure as fertilizer increased 0.3 times from 1982 to 1983, while the nitrate applied as fertilizer increased 2.5 to 3 times. The situation is similar over much of the midwestern corn belt in the U.S.

Generally, although nitrogen is taken up by the plant very quickly, the overapplication of nitrate fertilizers can adversely affect ground water quality. The critical factors influencing the leaching of nitrogen are the rate and timing of fertilizer application, the rate and timing of water applications, and soil characteristics. Findings by Fried *et al.* (1976) suggested that applications of nitrogen fertilizer at a rate above the crops' maximum yield needs will result in a great increase in the amount of leachable nitrogen. They state, "*. . . the key to minimizing agriculture's contribution to nitrate in ground water is the efficient use of fertilizer as indicated by the proportion of added nitrogen which is removed by the harvested portion of the crop.*" The timing of application also greatly affects the amount of nitrogen available for leaching. If the fertilizer is applied at the pre-emergent or at the post season stage, crop uptake (the major utilizer of the nitrogen) will be absent. The rate of water application is also a crucial factor. As more water is applied at one time, more nitrogen can be expected to percolate through the soil profile. "*The distance that nitrate is moved downward is a function of the water*

entering the soil, and the more rainfall that occurs, the deeper nitrate can be expected to move in a given soil" (Thomas, 1970). Also crucial is the timing of water applications. If a crop is irrigated excessively while the plants are small, the leaching of nitrate could result even if the amounts of nitrogen applied are not excessive (Campbell, pers. comm., 1982). In addition, if irrigation is done less frequently but in increasing amounts, more nitrate will leach deeper in the soil profile than if irrigation is done on a more frequent, lighter application schedule (Singh and Sekhon, 1979).

3.4.3 Nitrogen in the Soil Column

It has previously been noted that the texture (permeability) of soil greatly affects the rapidity and extent to which water will leach through the soil profile. Therefore, as an example, the potential for nitrate contamination of ground water under sandy soils is greater than for silt loam soils. In addition, the organic content of soil, its cation exchange capacity, and the aerobic and anaerobic conditions that are present, determine the extent of nitrification and denitrification taking place in the unsaturated and saturated zones.

In its downward movement from the surface, nitrate traverses the root zone, and both the unsaturated and saturated zones. In the root zone the crop will remove some of the applied nutrients present in the soil solution. A major factor limiting the amount of applied nitrogen that will leach below the root zone is the type of crop cover and the extent of its root system. The amount of nitrate that passes below the root zone is not necessarily the same concentration that will reach the ground water. This is due to dispersion and diffusion of the substance as it moves through the soil.

In the unsaturated zone nitrate tends to be stable. This is because the conditions generally existing in this zone limit anaerobic microbial activity which could lead to denitrification. Studies have shown nitrate concentrations within the unsaturated zone to be highly variable. One factor governing the nitrate concentration in this zone is the drainage or percolation volume. A second factor is the difference between nitrogen inputs and crop removal, and a third involves the soil profile characteristics or conditions which control denitrification.

Nitrate has a high mobility in ground water, with migration rates equivalent to 50 per cent of the velocity of the ground water carrying it (Jackson, 1982). The concentration of nitrate in the saturated zone can be reduced by dilution, and by either biochemical, or chemical reactions. A high soil organic matter content and low oxygen conditions promote denitrification. Denitrification can take place as nitrate reacts with various soil constituents to become reduced to nitrogen gas. (Hendry, McReady, and Gould, 1984). Bouwer *et al.* (1981) suggested that as a general rule approximately one quarter of the nitrogen applied to a crop will be lost by denitrification and other processes. If there is an absence of an adequate source of energy (organics) to permit bacterial denitrification to occur, none of the nitrate will be lost as gaseous nitrogen.

3.4.4 Ground Water Contamination

Singh and Sekhon (1979) referred to studies in Canada, the United States, Germany, Israel, and India which provided evidence of nitrate enrichment in ground water resulting from the application of nitrogen fertilizer to cropped land. They stated " . . . *investigations have indicated that nitrate is accumulating in the shallow ground waters of some irrigated areas with intensive agriculture using fertilizers.*" Recently, rising nitrate levels have been detected in the principal aquifers of Britain and leaching of agricultural soils was identified as the greatest source of nitrogen input to aquifers (Oakes and Young, 1981; Foster, Cripps, and Smith-Carrington, 1982). In various parts of Nebraska, high nitrate levels have also been discovered in ground water. They are believed to be a result of the application of nitrogen fertilizer to irrigated corn on permeable soils (Ferguson, 1980).

In Merrick County, Nebraska, extensive nonpoint nitrogen contamination of ground water was discovered (Spalding *et al.*, 1978). About 50 per cent of that county is underlain by ground water with nitrate-nitrogen concentrations 1.5 to 3.0 times above the 10 mg/L standard. Approximately 34 per cent of the county is planted in irrigated corn (Spalding *et al.*, 1978 stated " . . . *the absence of heterogeneous points of high and low concentrations and the large areal expanse of homogeneous nitrate-nitrogen concentrations within the 10 to 30 mg/L intervals are indicative of a widespread, relatively homogeneously dispersed source*"). A model was developed which attempted to correlate nitrate concentrations with soil drainage capacity and well densities in a given area. The nitrate concentrations in ground water under fertilized, irrigated land were found to be dependent on leaching by irrigation return flow. Nitrate concentrations were highest under the well drained soils which were being moderately to heavily irrigated.

Hallberg (1987) investigated nitrates in ground waters within the Big Spring Ground Water Basin in northeastern Iowa. That study showed that from the late 1960's to the present, as the difference between the amount of fertilizer applied and that taken up by crops increased, the nitrate concentration in the ground water increased threefold, from about 3 mg/L in the 1960's to 9 mg/L in 1982, and 10.1 mg/L in 1983. Baker and Laflen (1983) noted that nitrate nitrogen losses with subsurface drainage relate in nearly linear fashion to application rates exceeding 50 kg/ha.

Saffigna and Keeney (1977) examined nitrate concentrations in the ground water under the sandy soil of central Wisconsin. Nitrate in ground water was initially examined by sampling irrigation wells in the area. Twelve of the 29 sites sampled (40 per cent) had nitrate-nitrogen concentrations greater than 10 mg/L during the sampling period of July and August. Marked variations of nitrate concentration were found between wells. This was true even for wells which were adjacent to each other. Nitrate levels were also measured at a Wisconsin State University Experimental Farm. Concentrations were found to be consistently the lowest in wells adjacent to land which had received low levels of nitrogen fertilizer and in wells adjacent to land which had only recently been cropped. Levels were higher where the fields were in continual use and received large irrigation and fertilizer inputs. Saffigna and Keeney (1977) concluded that " . . . *nitrate in each well seems to reflect closely the agricultural practices in the immediate vicinity of the well.*" Their overall

conclusion was that irrigated crop land represented the major source of nitrogen found in ground water.

Timmons and Dylla (1981) examined the effect of two water application levels and two nitrogen fertilization methods on nitrogen leaching losses on sandy loam soil over a five-year-study period. Supplemental irrigation of 2.5 cm or 5.0 cm was applied to corn whenever there was a soil water deficit of 50 per cent. The soil water percolation at the 5.0 cm application rate was 78 per cent greater during the growing season and 61 per cent greater annually, than at the 2.5 cm level. The average annual leaching losses of nitrate from fertilized, irrigated fields were 17 to 62 per cent above the losses from fertilized, non-irrigated corn fields. The average annual nitrate-nitrogen loss for the 2.5 cm irrigation level was 80.95 kg/ha (average of the two treatment methods) compared to an average of 105.85 kg/ha for the 5.0 cm level. The 2.5 cm water applications produced yields comparable to the higher irrigation level and reduced the nitrate leaching losses considerably. The application of the liquid nitrogen rather than granular N decreased annual nitrate-nitrogen losses by 12 kg/ha, but only at the 5.0 cm irrigation level. This study revealed, as does that of Smika *et al.* (1977), that the loss of nitrate through deep percolation on soils receiving similar fertilizer inputs varies directly with water application rates.

An example of the effect of variations in amount and timing of N fertilizer applications is provided by Gerwing *et al.* (1979) for irrigated corn on sandy loam soil. Two amounts of urea nitrogen, 179 and 269 kg/ha, were applied at one time or in four separate applications. Irrigation rates varied from 2.5 cm a week early in the season to 5.0 cm a week later in the season. The concentration of nitrate in the soil solution at 1.5 metres increased almost immediately after irrigation began. Two months after the nitrogen was applied at the one-time application rates, the concentrations of nitrate-nitrogen peaked at approximately 127 mg/L in the soil solution for the 269 kg/ha rate and 98 mg/L for the 179 kg/ha rate. These peaks dropped off rapidly, followed by a similar pattern at the 2.4 m level. The concentrations were reduced by dilution and dispersion of the nitrate. By early September, the nitrate-nitrogen concentration in the aquifer increased 7 mg/L from a one-time application of 179 kg/ha. The 269 kg/ha one-time application resulted in an increase of 10 mg/L in the nitrate concentrations. The split applications of 179 kg/ha did not increase the nitrate-nitrogen concentration in the aquifer, while the 269 kg/ha split application had a "*minimal effect*" on the ground water nitrate-nitrogen concentrations (2 mg/L increase). When the 179 kg/ha of nitrogen fertilizer was applied at once, the crop recovered 30.4 per cent of the applied nitrogen. This compares with 52.4 per cent recovered in the split applications. Increased fertilizer use efficiency from split applications resulted in more nitrogen being used by the crop and less available for leaching. This, in turn, resulted in smaller nitrate concentration increases in the underlying ground water.

A model has been developed by Fried *et al.* (1976) for predicting the leaching of nitrogen to ground water. They have suggested that, over time, the soil will reach a steady-state level where the input equals the output. The nitrogen input to the soil-plant system is in the form of fertilizer, irrigation additions and natural additions. The leaching loss will equal the nitrogen inputs minus plant uptake, denitrified nitrogen and physical losses. Fertilizer use efficiency (the actual amount of nitrogen uptake by the crop) is seen as the key

factor affecting the amount of nitrogen subject to leaching in a given situation. This study indicated that after the maximum potential yield of a crop is reached, the amount of nitrogen subject to leaching increases dramatically with each successive increment.

3.4.5 Summary

This brief review has pointed out that widespread rising nitrate levels in ground water have been detected in various parts of the United States (e.g., Wisconsin and Nebraska) and Britain. The primary source of the nitrate was identified as nitrogen fertilizer combined with crop irrigation. It is probable that in western Canada a similar situation exists in many aquifers under irrigated lands. The studies reviewed indicate that as a result of what could be considered "*normal*" farming practices, nitrate contamination of ground water may take place. The rate and timing of fertilizer applications can have a significant input on nitrate losses from the root zone and ground water contamination. The rate and timing of water applications was also shown to significantly affect nitrate leaching losses. It is clear that the larger the fertilizer nitrogen and water applications are, the more nitrate enrichment of ground water can be expected. Fertilizer use efficiency by the crop plays a central role in the final amount of nitrate that leaches through the root zone. Split applications of fertilizer, in comparison with one time applications, increase the percentage of nitrogen taken up by the crop and greatly reduce deep percolation losses. An alternative to split applications is the use of slow release fertilizers which match nitrogen release with nitrogen plant needs (Skogerboe and Walker, 1981). When examining the potential for nitrate enrichment of ground water, three important factors must be considered: the volume of drainage water, the yearly excess nitrate available for leaching, and an estimate of denitrification. By maximizing fertilizer use efficiency and minimizing the leaching fraction of applied water, ground water contamination from nitrogen fertilizer under irrigated conditions can be significantly reduced.

4. SUMMARY AND CONCLUSIONS

The slow rate of ground water flow, the interaction between contaminants and the aquifer matrix, and the nature of contaminant transport all serve to make the cost of cleaning up contaminated aquifers prohibitive.

Within western Canada, the best quality ground water is most often found in surficial, unconfined aquifers. These aquifers are by their nature more susceptible to contamination by irrigation return flow than are deeper, confined aquifers. The threat to ground water quality is enhanced on the Canadian prairies where irrigation is practised, especially where the use of pesticides and nitrogen fertilizers is common. Because shallow aquifers are a very important source of water supply throughout the Prairies, the contamination of ground water supplies presents a very serious threat, and should be ranked highly on the environmental priority lists of federal and provincial governments. While this report is concerned principally with irrigated areas, the same holds true for non-irrigated areas where hydrogeological conditions are conducive to contamination of ground water from anthropogenic substances.

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The following persons, through discussion, have contributed toward the preparation of this report:

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